Methodology Development

A Simplified Model for Spatially Differentiated Impact Assessment of Air Emissions

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Abstract. We developed a simplified emission dispersion and exposure-assessment model designed to reflect the site-specific health impacts of air pollution in life-cycle impact assessment (LCIA). We proposed an EXposure Per Emission Coefficient (EXPEC), a dimensionless parameter representing the relative amount of pollutant inhaled per emission. EXPEC values were calculated for two typical source categories – electric power plants and road vehicles – on a concentric circle model. The EXPEC values were significantly different for different locations and source categories. We examined the application of EXPEC in a case study that compared the effects of emissions from electric and gasoline-engine vehicles. EXPEC is a useful tool for assessing spatially differentiated potential impacts.

Keywords: Air emissions; concentric circle model; electric vehicles, electric power plants; gasoline-engine vehicles; EXposure Per Emission Coefficient (EXPEC); LCIA; life-cycle impact assessment; spatially differentiated potential impact

Introduction

Air pollution is an issue of great concern. In Japan there has been evidence of respiratory damage caused by local air pollution near industrial complexes, and several cases involving possible health damage by automotive exhaust gases are being brought to trial. With the exception of the much-publicized global issues, environmental pollution is inherently location-specific. Most life-cycle impact assessment (LCIA) methodologies have neglected this feature by assuming so-called 'potential impact' rather than 'actual (or real) impact' (Owens, 1996). We attempted to develop a spatially differentiated impact-assessment method for air emissions by reflecting the realistic geographical distribution of sources and recipients of pollution.

1 Fate of Air Emissions and Their Spatial Contacts with Recipients

In any assessment of the potential or actual impacts of an emission, it is important to understand the fate of the emission in time and space, and to identify possible impact paths. Some substances take a long time to disperse and have impacts on a global scale. Examples of impact paths caused by these kinds of substances are climate change caused by greenhouse gases, and ozone-layer depletion caused by CFCs. For these types of impact paths, it is reasonable to assume that the contribution of a unit amount of emission to the impact is constant, wherever on the earth the emission occurred.

Acidification of air, water and soil by air emissions – one of the best-known impact categories in LCIA – takes places primarily on a regional scale. Under regional international agreements to cut down the emission of potentially acid substances, it may be reasonable, at least in a policy context, to assume that an impact will be of the same magnitude as its emission within a region. However, in reality, sources in different locations produce variably sized depositions to different sites because of non-uniform meteorological and geographical conditions. Moreover, different recipients of acid deposition have different sensitivities. Potting et al. (1998) discussed this issue and proposed some site-specific impact factors. In East Asia, long-range transport of acid substances is strongly dependent on the prevailing wind direction from west to east, so the relationship between the location of an emission and its deposition is highly controversial.

Toxic substances such as carcinogens are also emitted to air and reach recipients directly through inhalation and/or via other media such as food chains. As many of these substances are characterized by long lifetimes and cross-media movement, environmental fate and exposure models such as the Uniform System for the Evaluation of Substances (USES) have been applied to LCIA, as described by Guinée et al. (1996). Because of the complexities and uncertainties involved in cross-media transport of these substances, the location-specific conditions are not treated individually in the model, but are reflected merely in an averaged manner. Therefore, the exposure of populations near emission sites (such as chemical installations) is not fully taken into account.

Traditional air pollution, which is generated mainly by the combustion of fuels, is highly site-specific, because a population's exposure to the pollution usually takes place near the emission source within a relatively short time frame, that is immediately after the release of pollutants into the air. Both the stationary sources and mobile sources of this type of air pollution are located adjacent to residential areas, so the population's exposure to this pollution is highly influenced by its spatial relationship with the source of emission. For example, in a Swedish study on EPS by Steen (1996),

exposure assessment was implicitly used to determine characterization factors. The EPS's characterization factor was derived from a case study of exposure assessment in a specific urban area and should not be applied directly to other areas.

Photochemical air pollution or 'summer smog' is also highly site-specific, not only because of the non-uniform spatial distribution of the sources of precursors' emission, but also because of its dependency on local geographical and weather conditions.

In an effort to reflect the site dependency of the impacts of air emissions in LCIA, some researchers have tried to estimate real damages, and in doing so have demonstrated the usefulness of the site-specific approach. Among them, Spadaro et al. (1999) proposed a very practical approach for quantifying real impacts of air emissions by combining dispersion models and dose–response models. We proposed a similar approach to quantifying the size of total damages by combining the concentration of the pollutant (as predicted by a dispersion model) and the density of the receptor population (Moriguchi et al., 1998).

In this paper, we demonstrate our modeling and its results, focusing on the local population density near sources of air emissions. We assess the sensitivities of the population distribution surrounding different emission sources, and then apply our model in a case study of electric vehicles.

2 Modeling

2.1 Relationships among emission, ambient concentration and dose

The problems of time and space in relation to the fate of pollutants have been discussed within LCIA, particularly with regard to hazardous substances and their associated health risks. The time problem is called the pulse-flux or flux-pulse problem, and arises when an environmental model is used to estimate health risks. The space problem arises from the practical simplicity of LCIA, in which emissions from different sources are aggregated regardless of their locations. Although the notion of potential impact is practical and useful, the assumption that air pollution mixes uniformly in the earth's atmosphere is unrealistic when it is applied to local air pollutants such as particulate matter, NO_x, SO_2, etc. (Spadaro et al., 1999).

The output of inventory analysis in LCA is generally an emission (pulse), which means a quantity expressed as a mass. The input required for environmental models in the risk assessment field is emission per unit time (flux); flux is converted into health risk by the calculation of a concentration or exposure dose. Therefore, one line of thought is that a pulse should be converted into a flux in order to calculate health risk with a ready-made environmental model, and some solutions have been proposed in this context (Guinée et al., 1996). However, our alternative method of calculating health impacts of local air pollution is much simpler, as a similar method has been described by Spadaro et al. (1999).

In our approach, we tried to relate the emission mass directly to the exposure dose by introducing a coefficient representing the ratio of the exposure dose to the emission. This ratio, which we called the 'EXposure Per Emission Coefficient (EXPEC)', expresses the amount of emission inhaled by the local population surrounding the emission source relative to the total emissions of pollutants. Similar coefficients representing the ratio between potential dose and a unit emission have been proposed, as in the case of 'exposure constants', but they exclude population density and distribution, and they are often expressed with a time dimension (for example, per day) (Hertwich et al., 1999).

EXPEC can be determined by working out the site-specific population density and ground-level concentration of a pollutant, which are estimated from a dispersion model according to the location of the population relative to the emission source. Therefore, EXPEC is a dimensionless coefficient as follows:

$$\text{EXPEC} = \frac{1}{Q} \iint V \times C\ (x,y) p(x,y) dx dy \qquad (1)$$

where

V = inhalation volume per person per unit time [m^3/s/person]

x,y = locations of receptors relative to emission sources

C (x,y) = ground-level concentration of pollutants at a site (x,y) [g/m^3]

p(x,y) = population density at a site (x,y) [persons/m^2]

Q = mass of emission per unit time [g/s]

2.2 Direct relationship between emission and response

Once EXPEC has been quantified, it can be applied to convert the quantity of an emission in inventory to the absolute size of the health impact on a local population, using a dose–response relationship.

First, from eq. (1), Q × EXPEC [g/s] represents the total amount of pollutants inhaled by the local population per unit time under steady-state conditions of continuous emission and continuous exposure. Therefore, for any length of exposure, additional emission of the pollutant W [g] will increase the total inhaled dose in the population by W × EXPEC [g].

Second, this total inhaled dose can be converted to a population risk. For example, the hazard level of the carcinogens in the atmosphere is usually expressed as a unit risk (UR), representing the increase of risk probability for each 1 $\mu g/m^3$ rise in concentration. UR [case/person] is usually given as the risk over an exposure period of 70 years. Therefore, the total inhaled amount of pollutants I_t [g] corresponding to 1 $\mu g/m^3$ is as follows:

$$\begin{aligned} I_t\ [\text{g/person}] &= V\ [\text{m}^3/(\text{persons} \times \text{s})] \times 1\ \mu\text{g/m}^3 \times 86\,400\ [\text{s/d}] \\ &\quad \times 365.24\ [\text{d/y}] \times 70\ [\text{y}] \times 10^{-6}\ [\text{g}/\mu\text{g}] \\ &= 2209V\ [\text{g/person}] \end{aligned} \qquad (2)$$

I_t is the mass of pollutant that causes an increase in the cancer probability by the UR when inhaled by a single person. Therefore, when 1 g of a pollutant with the unit risk UR is inhaled, the probability of cancer in a person will increase by UR/2209V. Usually, the recipient of the impact is not a single person but a whole population, so the increase in risk

level is 'shared' by the population. Therefore, it is more appropriate to describe this situation as follows: when 1 g of the pollutant with the unit risk UR is inhaled by the population, the risk to that population will increase by UR/2209V [case]. Here we apply EXPEC. If 1 g of the pollutant is emitted in a given location, the population will inhale EXPEC g of pollutant, and the population risk will increase by EXPEC × UR/2209V [case]. Thus, by converting pollutant's concentration to its dose, the mass of the pollutant in emission inventory can be directly converted to the quantity of damage, without the pulse-flux problem.

A very similar calculation process was found in a US-EPA's study (Bouwes et al., 1997), in order to calculate 'indicators' representing the overall magnitude of the potential impact by a unit amount of chemical release from facilities registered in the Toxic Release Inventory. To calculate the indicators, they multiplied three components; the toxicity of chemicals, the quantity to which people are exposed, and the size of the population exposed to those chemicals.

Here, we may derive another coefficient, EXPEC divided by 2209V (the conversion factor between concentration to inhaled lifetime dose in the eq. (2)). We can more easily and directly calculate a population risk by this alternative coefficient and UR for air pollutants, and V can be canceled out from the calculation process. It is interesting to know that the alternative coefficient has a dimension of [person/m³], which represents population density weighted by air pollutant's concentration within the space for its dispersion. Though practical usefulness of this alternative coefficient should be further examined, we adopted our original idea of EXPEC because its concept can be more generally applicable.

2.3 Concentric circle model

Although eq. (1) can be applied to any case by using individual site-specific emission data and site-specific population-distribution data, it is not practicable to specify the locations of individual processes comprising a life cycle and to reflect them in the impact assessment. However, there is room to reflect the averaged characteristics of sources and the population distribution around them. For example, thermal-electric power plants with high stacks are located mainly in coastal areas away from cities, whereas automotive emissions often occur at ground level in densely inhabited areas. If a set of EXPEC values is prepared in advance for a typical emission source, it can be used immediately in LCIA without the need for exhaustive calculations with a dispersion model.

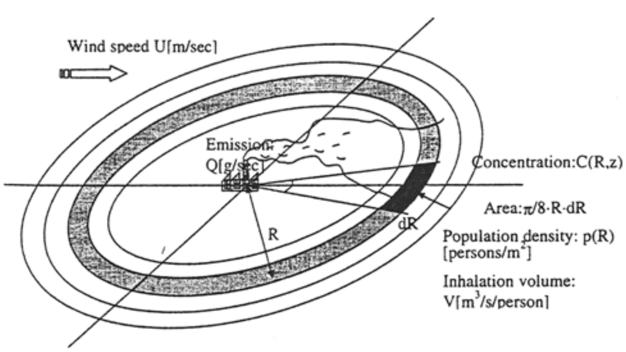

Fig 1: Illustration of emission and exposure on the concentric circle model. Emission: Q [g/s]; Wind speed: U [m/s]; Population density: $p(R)$ [persons/m²]; Area: $\pi/8 \times R \times dR$; Inhalation volume: V [m³/s/person]

In order to calculate real values of EXPEC for this purpose, we used a simplified concentric circle model.

This model (**Fig. 1**) puts an emission source in the center of concentric circles, and assumes that the concentration of pollutants changes only with the distance from the source. This means that we assume a hypothetical wind condition in which the incidence of pollution is the same from all wind directions. If this assumption is made, the calculations made with the dispersion model are greatly simplified.

With the Gaussian plume model in a polar coordinate system, the concentration at the leeward exposure point *C(R,z)* is as follows:

$$C(R,z) = \frac{1}{\sqrt{2\pi}} \frac{Q}{\frac{\pi}{8} R\sigma_z U} \left[\exp\left\{ -\frac{(z-He)^2}{2\sigma_z^2} \right\} + \exp\left\{ -\frac{(z+He)^2}{2\sigma_z^2} \right\} \right] \quad (3)$$

where

$C(R, z)$ = concentration of pollutant at exposure point [g/m³]
R = distance from point source to exposure point [m]
z = height of exposure point [m]
Q = emission intensity [g/s]
σ_z = dispersion parameter dependent on R($\sigma_z = \alpha R^{\beta}$)[m]
U = wind speed [m/s]
He = effective height of emission source [m]

This can be rewritten as follows where z = 0:

$$C(R,z) = \frac{16}{\sqrt{2\pi}} \frac{Q}{\pi R\sigma_z U} \left\{ \exp\left(-\frac{He^2}{2\sigma_z^2} \right) \right\} \quad (4)$$

When the population density is *p(r)* [persons/m²] and the inhalation volume of a human is *V* [m³/(person x s)], the inhalation volume per unit area and unit time is *p(r)V* [m³/(m² s)]. The exposure dose of a substance per unit area and unit time at the exposure point is calculated below.

Exposure dose rate at exposure point [g/(m² s)]
= inhalation volume per unit area and unit time [m³/(m² s)] × concentration [g/m³]
= *p(R)V* × *C(R, z)*

$$= p(R)V \frac{16}{\sqrt{2\pi}} \frac{Q}{\pi R\sigma_z U} \left\{ \exp\left(-\frac{He^2}{2\sigma_z^2} \right) \right\} \quad (5)$$

Therefore, the cumulative exposure dose per unit time [g/s] for all the population in a downwind direction is as shown below:

Cumulative exposure dose [g/s] =

$$\int_0^{R_{max}} \left(p(R)V \times \frac{16}{\sqrt{2\pi}} \frac{1}{\pi R\sigma_z U} Q \right) \left\{ \exp\left(-\frac{He^2}{2\sigma_z^2} \right) \right\} \frac{\pi R dR}{8} \quad (6)$$

By definition, EXPEC is as follows:
EXPEC = cumulative exposure dose per unit time [g/s] / emission per unit time [g/s]

$$\text{EXPEC [-]} = \sqrt{\frac{2}{\pi}}\frac{V}{U}\int_0^{R_{max}}\frac{1}{\sigma_z}\left\{\exp\left(-\frac{He^2}{2\sigma_z^2}\right)\right\}p(R)dR \qquad (7)$$

By solving eq. (7) with site-specific $p(R)$, H_e and the meteorological parameters (U, σ_z), site-specific values for EXPEC can be obtained. In this study, the sensitivity of the meteorological parameters was not analyzed; we considered only the sensitivity of $p(R)$ and H_e. The atmospheric stability was assumed to be neutral, and the wind speed U was assumed to be 3 m/s. The inhalation rate V was assumed to be 1.74 x 10^{-4} m^3/(person x s) (15 m^3/(person x d)). Integral calculus was performed for up to a 100-km radius from the center (R_{max} =100 km in equation (7)).

3 Calculation of EXPEC for Typical Sources

3.1 Materials and methods

In order to demonstrate the range of EXPEC values for different sources, we chose electric power plants and exhaust from road vehicles for inter-comparison.

Throughout the study we used a so-called 1-km grid system that covers the whole of Japan. Each grid actually covered 1/120° (30') of latitude and 1/80° (45') of longitude. In a north-south direction this was equivalent to 0.924 km, and in an east-west direction the length depended on the latitude (at 35°, the approximate center of Japan, the length of the side of the grid would be about 1.13 km). Results from the nationwide population census are available from the National Bureau of Census on this 1-km grid scale.

We specified the locations of 112 thermal-electric power plants, and then calculated the population density around each plant along concentric circles (see Fig. 1.) We then averaged the population densities around each plant, with weightings by fuel consumption in calorific value, to calculate the national average and averages for regional electricity suppliers. Weighting by fuel consumption was used as a surrogate for weighting by pollutant emission, because of the limited availability of emission data. We assumed the stack height of each power plant to be 200 m.

Vehicles produce emissions wherever they are used. Therefore, the value of EXPEC depended on where the vehicles were used. We calculated EXPEC for road vehicles as a national average and as averages for each of the 47 prefectures covering the whole of Japan by the following scheme. By combining the results from the national census of traffic on trunk roads and their location data by a digitized road map, we calculated the traffic volume (vehicle-km) in each 1-km grid for all of the grids where trunk roads existed in these prefectures. About 60-thousand 1-km grids were involved as sources. We calculated the population densities around all of these source grids along the concentric circles (precisely speaking, circles approximated by pieces of grids). We then averaged the population densities for each prefecture and for the nation as a whole by weighting for traffic volume. This weighting was a surrogate for weighting by emission. Though traffic emissions are usually modeled as line sources, we applied the same concentric circle model by substituting them by a series of point sources. However, sensitivities of EXPEC to local population distribution and to local dispersion patterns near roadways should be subject to further research.

3.2 Results

We obtained results for the following: changes in ground-level concentration of emissions with distance from power plants and road sources (**Fig. 2a**); changes in population density with distance from sources (**Fig. 2b**); and changes in cumulative exposure with distance from electric power suppliers and typical sites for automotive exhaust emission (**Fig. 2c**). Because emission from vehicles on roadways gives a much higher ground-level concentration of pollutants than emission from power plants with high stacks (see Fig. 2a), the total exposure of the population is higher for vehicle emissions, even if the population densities around the sources are the same. If vehicles are used in dense urban areas, the difference is even larger. Emissions from vehicles in a densely populated urban area like Tokyo had over 10 times as much impact as those from power plants in national locations (see Fig. 2c.)

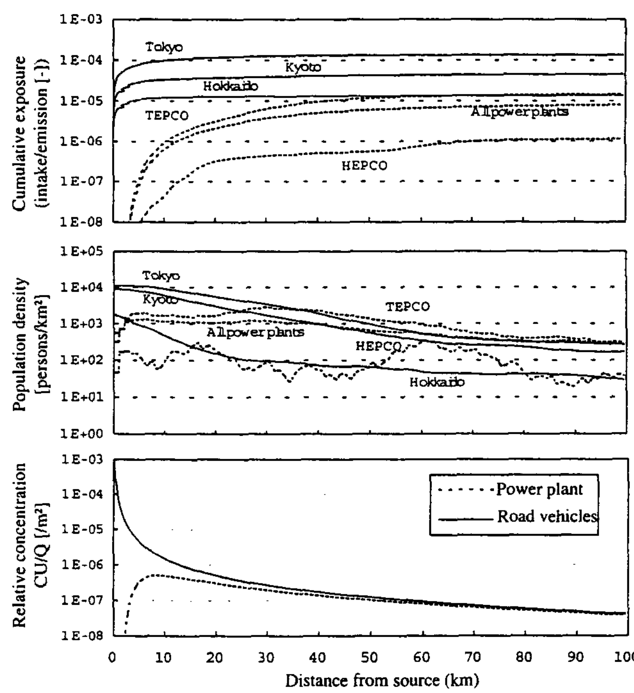

Fig. 2: Changes in (a) ground-level concentration, (b) population density and (c) cumulative exposure with distance from different sources; (a) Relative concentration CU/Q [-/m²]; (b) Population density [person/km²]; (c) Cumulative exposure (intake/emission [-])

There were significant differences in EXPEC values among different sources in different locations (**Table 1**), because of variations in the population distribution over the nation.

Table 1(a): Exposure per emission coefficients (EXPECs) calculated for electric power

Supplier	EXPEC (x 10^{-6})
National average	8.1
HEPCO	1.2
TEPCO	15.0
KEPCO	8.8

HEPCO = Hokkaido Electric Power Company
TEPCO = Tokyo Electric Power Company
KEPCO = Kansai Electric Power Company

Table 1(b): Exposure per emission coefficients (EXPECs) calculated for vehicle emissions by prefectures

Code	Prefecture	EXPEC (x 10^{-6})	Code	Prefecture	EXPEC (x 10^{-6})
0	National	34.5	24	Mie	15.8
1	Hokkaido	13.6	25	Shiga	19.6
2	Aomori	8.0	26	Kyoto	45.8
3	Iwate	5.9	27	Osaka	96.1
4	Miyagi	14.3	28	Hyogo	41.3
5	Akita	5.7	29	Nara	30.7
6	Yamagata	8.5	30	Wakayama	15.4
7	Fukushima	8.4	31	Tottori	8.1
8	Ibaraki	18.8	32	Shimane	6.0
9	Tochigi	15.3	33	Okayama	14.4
10	Gunma	19.2	34	Hiroshima	20.9
11	Saitama	59.7	35	Yamaguchi	10.6
12	Chiba	46.7	36	Tokushima	11.5
13	Tokyo	135.2	37	Kagawa	15.1
14	Kanagawa	79.1	38	Ehime	12.6
15	Niigata	9.8	39	Kochi	10.5
16	Toyama	11.2	40	Fukuoka	28.1
17	Ishikawa	11.3	41	Saga	12.6
18	Fukui	9.2	42	Nagasaki	14.0
19	Yamanashi	17.0	43	Kumamoto	12.7
20	Nagano	9.0	44	Oita	12.0
21	Gifu	16.8	45	Miyazaki	8.9
22	Shizuoka	19.6	46	Kagoshima	9.6
23	Aichi	40.8	47	Okinawa	23.9

3.3 Case study: Application of EXPEC in a comparison of the impacts of emission from electric and gasoline-engine vehicles

It is sometimes said ironically that the electric vehicle is not a Zero Emission Vehicle (ZEV) but an Elsewhere Emission Vehicle (EEV), because the power plant supplying electricity for the vehicle itself produces emissions. This is qualitatively true, and it is essential to consider such indirect emissions in LCA. According to our estimates, the emission of traditional air pollutants such as NO_x per unit vehicle-kilometer is approximately the same from recent gasoline-engine vehicles with modern catalytic converters as it is from prototype electric vehicles for which electricity is supplied by a Japanese fuel mix. However, LCA should go a step further, not only to compare inventories of the emission, but also to compare the impacts of direct emissions from engines of conventional vehicles and indirect emissions from electric vehicles. The national average EXPEC value for road vehicles is about 4 times the national average EXPEC value for electric power plants (see Table 1). When vehicles are used in densely inhabited sites such as Tokyo, the difference in EXPEC is much more, at most 10 times. In contrast, the EXPEC value for road vehicles in sparsely inhabited prefectures is even smaller than the national average EXPEC value for electricity suppliers.

We made trial calculations of the absolute sizes of health impacts from the use of gasoline-engine vehicles (GV) and electric vehicles (EV) by applying EXPECs at Hokkaido, at Tokyo and in the national average. We considered the carcinogenic effects of 4 heavy metals (cadmium, arsenic, chromium and nickel) emitted by coal-fired power plants as impacts from EVs, and the carcinogenic effects of benzene as an impact from GV. Inventories of heavy metal emissions per kWh for respective regional electricity suppliers, as estimated by Matsuno et al. (1998), were multiplied by the fuel-efficiency of the electric vehicle IZA (0.214 kWh/km) to estimate the heavy-metal emissions (in grams) per kilometer driven. The benzene emission from GV was assumed to be

Table 2: Estimated impacts of carcinogens emitted per kilometer driven by electric vehicles (EV) and gasoline-engine vehicles (GV)

	Carcinogen	Emission/kWh [g/kWh]	Emission/km [g/km]	UR [/(g/m³)]	Emission/km x UR	EXPEC	Population Risk[a] [case/km]
EV with National average electricity	Cadmium	1.00E-07	2.14E-08	0.0018	3.85E-11		
	Arsenic	2.00E-06	4.28E-07	0.003	1.28E-09		
	Chromium	2.00E-06	4.28E-07	0.04	1.71E-08		
	Nickel	9.00E-05	1.93E-05	0.0004	7.70E-09		
	Total				2.61E-08	8.10E-06	5.51E-13
EV with electricity by HEPCO	Cadmium	9.00E-07	1.93E-07	0.0018	3.47E-10		
	Arsenic	2.00E-05	4.28E-06	0.003	1.28E-08		
	Chromium	1.00E-05	2.14E-06	0.04	8.56E-08		
	Nickel	1.00E-04	2.14E-05	0.0004	8.56E-09		
	Total				1.07E-07	1.20E-06	3.35E-13
EV with electricity by TEPCO	Cadmium	4.00E-08	8.56E-09	0.0018	1.54E-11		
	Arsenic	8.00E-08	1.71E-08	0.003	5.14E-11		
	Chromium	4.00E-06	8.56E-07	0.04	3.42E-08		
	Nickel	8.00E-05	1.71E-05	0.0004	6.85E-09		
	Total				4.12E-08	1.50E-05	1.61E-12
GV in national average	Benzene		0.005	7.80E-06	3.90E-08	3.45E-05	3.50E-12
GV in Hokkaido	Benzene		0.005	7.80E-06	3.90E-08	1.36E-05	1.38E-12
GV in Tokyo	Benzene		0.005	7.80E-06	3.90E-08	1.35E-04	1.37E-11

[a] The population risk was calculated as emission/km x EXPEC x UR/2209V (V = 1.74E-4m³/s/person)
UR = unit risk; V = inhalation volume

0.005 g/km. We then multiplied the unit risk of the respective carcinogens and EXPECs for respective locations by the emissions per kilometer driven (Table 2). In this example, the emission of carcinogens per kilometer driven weighted by unit risk without EXPEC application was of the same order for EV and GV. However, when we applied EXPEC, we found that EV had a smaller impact than GV in terms of these selected carcinogens. In Hokkaido, where the share of coal-fired power plants is relatively high and the population density is low (except in limited urban areas), the emission of carcinogens weighted by unit risk was 2.5 times higher for EV than for GV, but the impact estimated by EXPEC was 4.5 times lower for EV.

This example should be interpreted carefully, because we did not cover all typical and significant pollutants of GV and power plants, but assessed the impact of only a limited number of carcinogens, of which unit risk is known and of which inventory data is available. Therefore, the results do not demonstrate the comparison of overall health impact of EV and GV. Nevertheless, we may draw the following observation from the case study.

The advantages or disadvantages of EV compared with GV are influenced significantly by the environmental performance of electric power plants, which typically have different CO_2 emissions from different fuels. From the viewpoint of the population's health risk from local air pollution, we found that the location of EV use was crucial. The use of EV in decreasing the health risks of local air pollution has much larger advantages in densely inhabited areas than in sparsely inhabited areas.

We must also pay careful attention to issues of intra-generation and inter-region equity. In this study we simply accounted for the number of people affected and neglected the societal implication of the spatial transfer of damages. Residents around power plants in suburbs and rural areas have to receive more risk from increased emissions, whereas residents in the metropolis can benefit from reduced risk by decreased emissions.

4 Conclusions and Future Perspectives

LCIA has been dealing mainly with the potential impacts of air pollution and has been reluctant to assess actual impacts, insisting that these should be dealt with by risk assessment. Japanese LCA case studies have principally investigated global emissions of pollutants such as CO_2, and there have been few opportunities to consider the local impacts of emissions. However, a larger number of substances such as carcinogens are now included in LCI, and more discussions will be raised to improve the reality of impact assessment.

It is important that impact-assessment methodologies are kept simple and feasible. In this context, authors do not intend to assess the actual impacts, but what we may call 'potential, but realistically differentiated impacts' or 'spatially differentiated potential impacts'. The EXPEC we have proposed in this paper is simple, but it can reflect the geographical characteristics of sources of local air pollutants and the population characteristics of their recipients.

In our empirical results we found large differences among source categories in assessed impacts per unit emission of local air pollutants. Emissions from vehicles in metropolitan areas have more than 10 times as much impact as the same mass of emissions from electric power plants. Such results suggest the merits of using electric vehicles, particularly in urban areas.

The EXPEC values in this paper were calculated by a very simple dispersion and exposure model, but they can be improved by using more sophisticated dispersion and exposure models. We calculated these values for only two typical source categories of air emissions (electric power plants and vehicles). Additional high-priority default source categories need to be identified. The iron and steel industries, petroleum refineries and the petrochemical industry are priority candidates, first because they are large contributors to air emissions, and second because the locations of these industries are very specific, as they are concentrated in a limited number of industrial complexes.

Although we discussed only local air pollutants, the concept of EXPEC can be extended to other pollutants in other media. We dealt with exposures taking place near sources over short time periods, scenarios for which EXPEC is eminently suitable. More sophisticated approaches will be needed to assess the impacts of persistent emissions via multi-media exposures.

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